Comparison of stream macroinvertebrate assemblages in canyon ecosystems of the Blue Mountains (Australia) with and without recreational traffic: a pilot study in impossible terrain

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ABSTRACT

In Australia, the adventure sport of canyoning occurs predominantly in the protected areas of the Greater Blue Mountains World Heritage Area, 50 km west of Sydney. It involves travelling through narrow, deep gorges using a combination of walking, abseiling, wading, rock scrambling and/or swimming through the canyon streams. The sport's popularity is reported to have increased substantially over time, causing concern for the sustainability of these fragile ecosystems. To investigate visitor impacts, benthic macroinvertebrate assemblages and water quality were compared among four canyons subject to high visitation (average 20 – 100 visits per week) with those in four unvisited/low visitation canyons (0 - 5 visits per week). Comparison was also made between impacted and non-impacted areas within two of the high visitation canyons. Sampling was undertaken over two canyoning seasons (early spring to late autumn).

No significant relationship between macroinvertebrate assemblage composition or water quality was observed due to visitation level within and among canyons, although there were inherent differences among canyons. Water quality was consistent with pristine conditions in all locations. Macroinvertebrate assemblages included a range of pollution sensitive taxa (e.g., Ephemeroptera represented by 42.3% of all animals). Although family richness was concentrated within a few orders (84.2% of families were represented by 50% of orders), most families were represented by 2 Recognisable Taxonomic Units, 47.4% of these were represented by a single taxon.

We concluded that, at current visitation levels, there is no measurable impact on the predominant animal component of these ecosystems. This finding was contrary to the expectations of the land managers of this area.

Key words: wilderness management; recreational impacts, freshwater aquatic environment, water quality, trampling, upland streams.

Introduction

Disturbance may be defined as "any relatively discrete event in time that is characterised by a frequency, intensity and severity outside of a predictable range that disrupts ecosystem, assemblage, or population structure and changes resources or the physical environment" (Resh *et al*. 1988, p. 433). Disturbance may therefore be considered an *event* that (a) changes parameters to outside the range normally experienced by the organism, assemblage, or environment in question, and not the effect flowing from such an event, and (b) whose occurrence is *unpredictable*. This latter factor is critical because organisms may be adapted to predictable seasonal fluctuations, for example changes in discharge rates, temperature, dissolved oxygen and/or other parameters (Reynolds 1983; Resh *et al*. 1988; Brewin *et al*. 2000).

Effects of disturbances on assemblages and ecosystems may differ according to type. Bender *et al*. (1984) proposed a distinction between a *pulse* disturbance, whereby application of a relatively short-term disturbance causes a temporary alteration in parameters, such as abundance or density of an assemblage or selected species, after which there is recovery to the former state, and a *press* disturbance that continues for a longer period and results in a more prolonged change in natural parameters. Typical examples of pulse events are chemical spills in rivers that are rapidly diluted (Allan 1995; Angradi 1999), or floods (Townsend 1989), compared with *press* events such as forestry clear felling or long-term exposure to sewage effluent (Growns and Davis 1991; Chapman *et al*. 1995; Wright *et al*. 1995). A third

type, *ramp* disturbance, whereby intensity increases over time may also occur, for example drought (Lake 2000; Boulton 2003) or seasonal recreational activity (Hardiman 2003).

Freshwater environments, especially lotic ecosystems, have had limited research in comparison to terrestrial or marine environments, and knowledge of their biotic and abiotic processes is correspondingly less well known, particularly regarding mobile organisms (Yount and Niemi 1990; Downes and Keough 1998). Drying of stream channels in lotic environments normally occurs gradually, and this allows time for behavioural responses. Where droughts are predictable, many species have evolved life history or behavioural characteristics that enhance their survival and recovery from such periodic episodes. Where adaptation or avoidance has not occurred, organisms may become locally extinct and, if the drought is widespread, recolonisation may be delayed (Yount and Niemi 1990; Boulton 2003).

Most research on disturbances in lotic systems has concentrated on the effects of floods and spates as structuring forces on assemblages within such ecosystems. If flooding is regular and predictable it is not disturbance, for example, flooding caused by monsoonal rains in Nepal result in predictable change in macroinvertebrate assemblages (Brewin *et al.* 2000). This supports the view that ecological response reflects an adjustment to predictable, seasonal changes in flow. Conversely, unpredictable, severe floods and/or droughts can cause disturbance (Molles 1985; Flecker and Feifarek 1994). Many organisms rely upon periodic flooding for reproduction, for example, the life cycle of fishes in tropical floodplain rivers, such as the Amazon, are closely synchronised with events associated with the annual flood regimes (Flecker and Feifarek 1994), as are many species of fish in Australian ecosystems (Reynolds 1983; Geddes and Puckridge 1988). Abundance of stream invertebrate fauna, especially aquatic insects, is often strongly synchronised with changes in food and habitat availability caused by fluctuations in water level, especially in the northern hemisphere (Hart 1985). However, although a considerable body of research of flood effects on macroinvertebrate assemblages has been accumulated, both their initial response and their subsequent recovery still appear complex and not clearly defined. Understanding these impacts on mobile organisms in such systems is lacking (Ream 1980; Sousa 1984; Kuss *et al.* 1990; Yount and Niemi 1990; Downes and Keough 1998).

Author observation (Hardiman 2003), together with documented canyoner deaths in flash floods, show that sudden, severe spates are a frequent and unpredictable feature of most headwater streams in canyons of the Blue Mountains, Australia. Although the likelihood of changed frequency of such spates is somewhat predictable within season, their actual occurrence in a specific canyon is unpredictable. Although spates may be of short duration (e.g., 1 day) their effects can be sudden and dramatic. For example, within a few minutes, water levels may reach more than 5 m above a nominal water level of less than 1 m, and the stream may go from a trickle to substantial velocity. In the narrow (typically <5 m wide), confined canyon environment such forceful disturbance may move large boulders, weighing many tons, downstream. Associated substratum scouring, movement of entire trees and other debris associated with the flood waters (e.g., considerable loads of soft sediment) may result in the debris being deposited downstream.

Unlike intermittent streams in arid zone canyons (e.g., Colorado Plateau, USA), where canyons can remain dry outside of the impact of infrequent flash floods, those in the Blue Mountains are perennial, although their flow can be greatly reduced or even halted, ultimately leaving a chain of pools in some canyons during extended drought. However, even when surface flow fails, hyporheic flow and/or deeper underground seepage through rocks usually continues, albeit at a slow rate. Blue Mountains canyons cannot therefore be considered to be subject to harsh or prolonged drought.

Apart from spates, the only other potential natural restructuring influence on the canyons is fire. Wildfires are frequent in the Blue Mountains, but as the aquatic sections of canyons tend to be between deep, narrow, rocky walls, fires seldom have a direct effect on these aquatic ecosystems although the burnt debris may enter streams as runoff from the catchment. This debris may temporarily increase allocthonous input that is eventually washed downstream.

Blue Mountains canyons are within largely undisturbed wilderness, protected as part of the Greater Blue Mountains World Heritage Area (NPWS 2001). They are distant from the effects of urban centres, industrial areas, and even private dwellings. There has been no clear-felling of vegetation within canyon headwater catchments. None of the canyons has vehicle access to canyons, and the only access requires bushwalking, usually over several kilometres, often on ill-defined tracks. The canyons are therefore effectively unaffected by human activities, except for the pedestrian recreational traffic of canyoning.

The effects of recreation activities on aquatic environments are generally the least understood aspect of 'carrying capacity', probably because change in such environments tends to be less immediately obvious than in terrestrial environments. Potential impacts may be of two types: anthropogenic pollution or physical disturbance. Research to date has tended to be focused on lentic ecosystems such as lakes or reservoirs, mostly linked to the concentration of use, typically during summer when the aquatic environment is most stressed due to factors such as low flow rates (Kuss *et al.* 1990).

Research has also been undertaken on motorboating impacts such as from wash, turbulence and turbidity, propeller action, and/or pollution from outboard motors and sewage. Other impacts associated with shore-based activities include angling and swimming. Research into recreational impacts on freshwater aquatic systems, especially in lotic systems in remote areas, is sparse (Ream

1980) and, within this context, the plants are relatively better known (e.g., Pickering and Hill 2007) than animals (the latter mostly on wildfowl and/or fish). However, in rivers used for canoeing, benthic invertebrates may be disrupted or locally eliminated in shallow zones (Liddle and Scorgie 1980).

In contrast to aquatic ecosystems, the direct physical impact of recreation on terrestrial systems (in the form of trampling) has received moderate research, particularly associated with soil erosion (Kuss and Morgan 1980, 1984; Calais and Kirkpatrick 1986), vegetation morphology, and its response (Sun and Liddle 1993; Cole 1995 a, b) and recovery (Kuss and Hall 1991). Within aquatic systems, research on trampling effects has been largely restricted to marine intertidal habitats, such as rock platforms (Keough and Quinn 1998), mangrove swamps (Ross 2006) or coral reefs (Kay and Liddle 1989).

In the same manner that spates can be considered a form of disturbance, recreational activity in canyons may be viewed as disturbance. This is because most visits are made during summer, and the volume of traffic (i.e., number and size of groups traversing the canyon) is not predictable. Individual visits would constitute pulse disturbances. However, if such visits were made by many people and/or occurred frequently (which is the case in easy to access, popular canyons), the effect may be equivalent to a press disturbance or, if increasing over time, a ramp disturbance. Under the Reice *et al*. (1990) definition, canyoner visit disturbances would be episodic (i.e., a canyon's probability of being disturbed during a particular time interval is independent of all other canyons at the same time).

Initially developed as an extension to bushwalking (hiking), the recreational activity of canyoning is now a recognised sport that involves a combination of bushwalking (hiking), abseiling (rappelling), wading, swimming and rock scrambling. Since the first recorded Australian canyon traverse in 1961 (Jamieson 2001), it has become a mainstream adventure recreational activity, with participants undertaking trips with commercial tour groups, outdoor clubs and/or as private groups and (more rarely) as individuals. Owing to the 'composite' nature of the sport (e.g., wetsuits, ropes and abseiling devices used are common to a number of sports), accurate estimates of participation levels are difficult to assess. Anecdotal evidence suggests, however, that the sport has grown rapidly in recent years in many countries, evidenced by active canyoning communities in the USA (Canyoneering Association 2008), Europe (European Canyoning Commission 2008) and Australia (Oz Canyons 2008). Within Australia, the sport is predominantly a Blue Mountains phenomenon due, at least in part, to the area's diversity of destinations and mild climate.

Concern by the park management agency over the apparent growth in canyoning in the Blue Mountains, and its potential impact on the fragile environment of these upland slot valleys (canyons) and their catchments, is evidenced by a comparison between the 1988 Blue Mountains National Park Draft Plan of Management (NPWS 1988), which makes negligible mention of the sport, and the 2001 Plan of Management (NPWS 2001), where extensive discussion is devoted to the impacts of canyoning and proposals were documented for major changes in park management policy to combat associated perceived impacts. However, because there are limited data available on the biological ecosystem processes in these largely unstudied environments, policy was largely based on subjective assessment.

There is also a lack of comparative data. This is because past Australian visitor-impact studies in protected areas have tended to focus on trail damage, particularly soil erosion and/or vegetation damage, from activities such as walking, trail-bike riding and horse riding (e.g., Calais and Kirkpatrick 1986; Goeft and Alder 2001; Newsome *et al*. 2002). No studies have focused on the environmental impacts of recreational activities in aquatic canyon ecosystems, except for the studies of our group (eg. Hardiman 2003; Wolfenden 2005). Potential canyoning impacts include trampling of the streambed, human waste and/or personal-use propriety products, and littering. In this paper we assess whether the current level of recreation activity is within the natural parameters of the canyons' ability to absorb the impacts, and we consider whether current visitation levels are ecologically sustainable. Using streambed macroinvertebrates and water quality as surrogates for ecological health, we tested the null hypothesis that there is no difference in macroinvertebrate assemblages or water quality among canyons due to high compared to low visitation.

Site description

This study was undertaken in the Blue Mountains National Park, located 50 km west of Australia's largest city, Sydney (Fig. 1). The climate of the area is cool temperate, with cool winters and mild summers. Average annual precipitation is approximately 1300 mm, with most rain falling late in the austral summer and in early autumn. Rainfall in the first canyon season of sampling (October 1st 1999 – March 31st 2000) was substantially less (777.5 mm) than in the second season (October 1st 2000 – March 31st 2001; 1125 mm). Temperature over the two years of the study was similar (Bureau of Meteorology 2001).

The Blue Mountains comprises a deeply dissected plateau covering 247,000 ha. Together with other national parks to the north and south, this complex of national estate areas forms part of the Greater Blue Mountains World Heritage Area. Although carrying the description of 'mountain', the highest point is approximately 1100 m above sea level. There are at least 400 canyons known in the region, generally located within a range of 600 – 800 metres above sea level, within the incised headwater valley region of waterways (Townsend 1989).

Underlying rock is generally soft, readily eroded, quartz lithic sandstones of the Triassic Narrabeen group. Rock types within this group include 'Burra Moko Head' and 'Banks Wall' (Department of Mines 1966). Water erosion

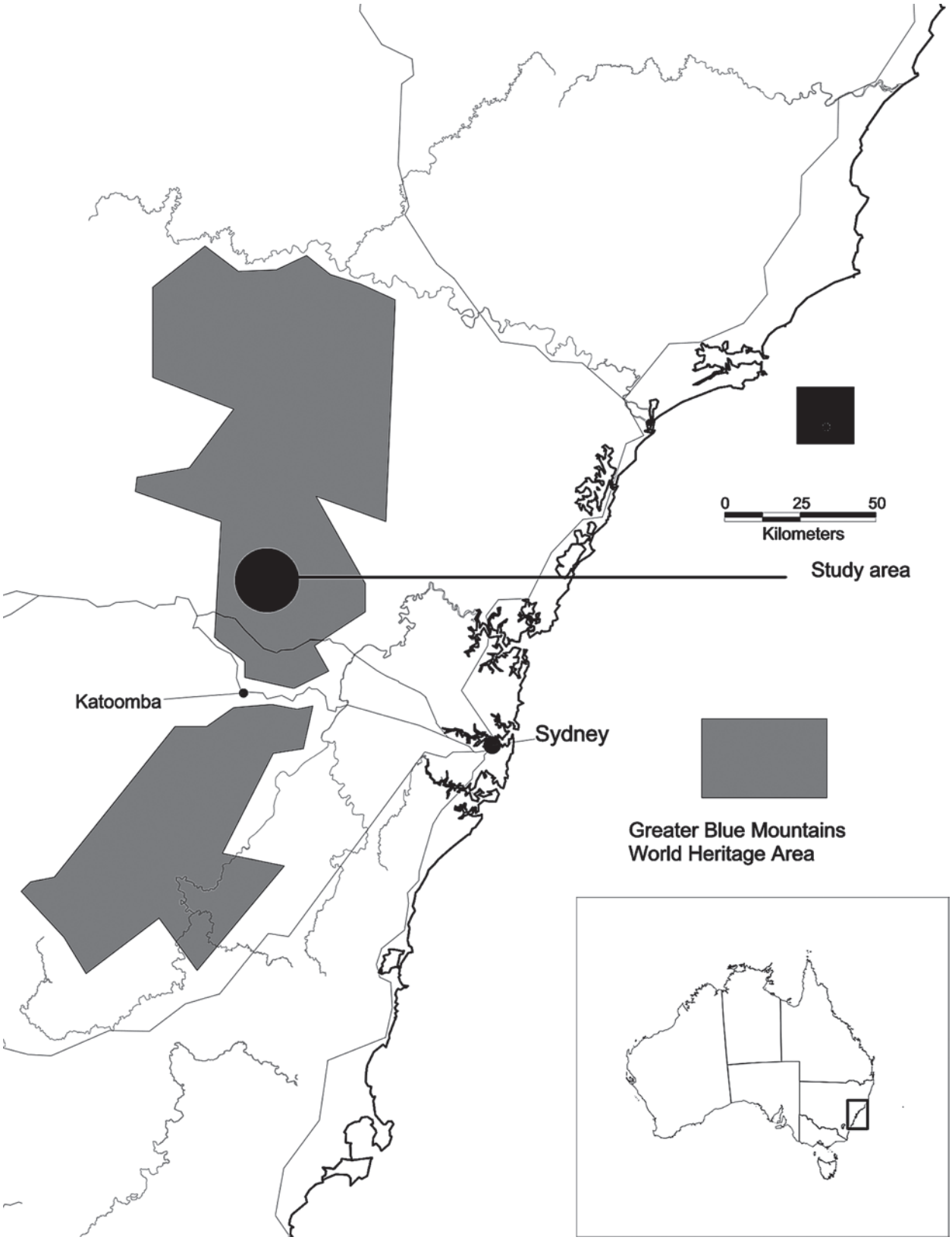

Figure 1. Location of the study area within the context of the Greater Blue Mountains World Heritage Area and the sites relation to Sydney, Australia's largest city.t

has formed narrow, dark, water-filled passages between sheer rock walls (i.e., canyons). Canyon streams are typically 4th order or lower (*cf*. Strahler 1957), with a dominant substratum of small to medium cobbles, with some stretches of sand, gravel, exposed bedrock and boulders. Although regimes may vary between individual locations, the streams are well aerated and can best be described as 'perennial flashy', having low flow predictability and possessing benthic macroinvertebrate assemblages strongly structured by abiotic processes (Allan 1995).

The canyons are surrounded by steep, rocky geology with natural bushland on poor low-nutrient soils, typically Open Forest or Open Heath associations, with a dominant canopy of *Eucalyptus, Leptospermum* and *Angophora* species (Benson and Keith 1977). Vascular vegetation is sparse to negligible within canyons, due largely to the lack of anchoring material and deep shading from the narrowness of the canyons within often effectively vertical walls.

Canyon ecosystems are considered fragile; either physically in terms of rock fragility (especially Burra Moko Head 'pagoda' rock formations that are friable and easily cracked), and/or biologically in terms of flora and fauna sensitivity. Although no detailed published species lists are available, canyons may occasionally harbour threatened endemic mammals (e.g., platypus *Ornithorhynchus anatinus*), many birds (including Superb Lyrebird *Menura novaehollandiae*), snakes and invertebrates (including crayfish *Euastacus spp*., glow worms, spiders) however, there is a lack of predators of invertebrates, such as carnivorous crayfish or fish present. Owing to its size and remoteness, the area has had limited research, but the potential for canyons to preserve rare species is demonstrated by the 1994 discovery of the Wollemi Pine *Wollemia nobilis*, a taxon considered extinct for 500 million years (Woodford 2000).

Methods

Site selection

Canyons sampled were all within the same catchment and biome, and located on streams of generally similar size, gradient, substratum, riparian vegetation and altitude. All were within essentially pristine environments, and not subject to urban, industrial or agricultural pollution. It was therefore assumed that any difference in faunal assemblages and/or water quality among sites was due to exposure to differing levels of recreational activity.

Four 'high-visit' canyons (Rocky Creek, Du Faur Creek, Serendipity, Bowen Creek) were compared with four 'low-visit' canyons (Bell Creek, Yarramun Creek, Hobnail Canyon, Nosedive Canyon; Table 1). In the absence of quantified data on canyon visitation, site selection was initially based on the senior author's personal experience and advice from the management agency, and subsequently confirmed by an on-site survey (Hardiman 2003).

The canyoning 'season' begins in October (austral spring), peaks in summer during January and tails off at the end of March (austral autumn), with visit levels correlated with water temperature (Hardiman 2003). Putative impacts from canyoning activity on streambed macroinvertebrate assemblages were, therefore, expected to be highest in mid-summer with cumulative impacts greatest in autumn. Sampling was undertaken five times in each of the eight canyons: October 1999 (spring), March 2000 (autumn), October 2000 (spring), December/January 2000/2001 (summer) and March 2001 (autumn). Sampling planned for summer 2000/2001 was cancelled due to a serious researcher injury in the field. In the last four sampling periods an upstream, low-visited section was added within two of the high visit canyons (Rocky Creek and Du Faur Creek) as a within-canyon control (topography did not permit this for the other two high-visit canyons). A total of 10 sampling sites, split between eight canyons, with quantified visit levels for each site, were sampled (Table 1). Canyon sampling order was re-randomised for each sampling period, and in each sampling period sampling was always completed within four weeks.

Sampling was restricted to riffle zones to maximise standardisation of sample sites. These were chosen at random during the first sampling period from among locations fitting the desired habitat characteristics. At each site (canyon) three riffle zones (replicates), at least 50 m apart, were sampled in riffle zones (< 50 cm depth, > 5 m long). Restricted entry/exit points and a lack of suitable riffle zones within the canyons limited freedom to sample and thus a Repeated Measures Design was employed (Green 1993; Keough and Quinn 1998), where time was the repeated (within-subject) factor and canyons were nested in visit level.

Table 1. Name and location of the ten sampling sites in the eight canyons, together with mean weekly visits. References: [1] Jamieson (2001); [2] Central Mapping Authority of New South Wales, topographic maps 1:25,000 scale. Sheets 8931-2-N (Rock Hill), 8931-2-S (Wollongambe), 8930-1-N (Mount Wilson); [3] includes visitors passing both ways through this canyon (all other canyons trafficked one way only).

Canyon name[1]	Map reference location of sampling sites[2]	Visit level	Mean weekly visits
Rocky Creek	High-visit sites: 481133 – 486133; Upstream low-visit sites: 475124 – 480130	High Low	99.4[3] 0.0
Du Faur Creek	High-visit sites: 531878 – 536894; Upstream low-visit sites: 529875 – 531880	High Low	28.0 0.0
Serendipity	566906 – 564915	High	36.4
Bowen Creek	578876 – 584875	High	18.9
Bell Creek	503895 – 516898	Low	3.4
Hobnail	553833 – 554848	Low	2.8
Nosedive	527052 – 526046	Low	0.3
Yarramun Creek	513943 – 531952	Low	0.0

Collection of data

Sampling equipment used was lightweight and waterproof as it was carried in backpacks and transported while swimming, abseiling, wading and walking for long distances, often over untracked terrain. On reaching a sampling location, physicochemical factors (i.e., water temperature, turbidity, dissolved oxygen, conductivity, salinity and pH) were first collected, and this was followed by the collection of macroinvertebrates. Although researchers necessarily travelled downstream (most canyons require abseiling down waterfalls), sampling sites were first carefully skirted and then kick sampled by moving back upstream through the water in a zig zag fashion, ensuring that all micro-habitats within a site were sampled within the 5 minute sampling period. Animals were collected in a kick net made of 250 mm mesh with a square mouth opening of 30 x 25 cm which was held vertically in the current, with the mouth in contact with the substratum facing upstream. Samples were emptied into a plastic sorting box and picked for 5 minutes, and then, to collect representative samples of debris potentially harbouring cryptic animals, four random grabs (eyes closed) with narrow-nosed forceps were collected. Animals and debris were then stored separately in a 5% formaldehyde solution in 50 ml plastic jars for transport back to the laboratory.

In the laboratory, animals were sorted from debris under a dissecting microscope illuminated by using a four dioptre 'Maggylamp' magnifying glass lamp for a standardised sorting time of 30 minutes before being preserved in 70% ethanol. Animals were subsequently identified to order, family and Recognisable Taxonomic Units (RTU). All benthic (streambed-dwelling) invertebrates were included in the analysis, with the exception of Oligochaeta and Turbellaria (worms and flatworms). These were excluded owing to difficulties of reliable identification, even to family level.

Analysis of data

Animal data were analysed by order, family and RTU using both univariate and multivariate techniques. Multivariate analysis was employed using the PRIMER V5 software package. Non-metric multidimensional scaling (NMDS) was performed on the (rank) similarity matrices of associations between all pairs of sample collections, computed using the Bray-Curtis measure. Interpretation of NMDS ordination plots presented here was via the relative distances of replicates on the plot: the closer together they are the more similar the invertebrate composition of the replicates (Clarke 1993; Clarke and Warwick 1994).

Employing replicated data from each canyon at each time of sampling, two-dimensional NMDS plots were used to investigate differences within- and among- canyons. NMDS plots using untransformed, square root, log x + 1 and fourth root transformed data were initially constructed and compared to assess whether any difference in transformation could distinguish differences in assemblage composition. When no substantial difference was evident, square root transformation was subsequently adopted as the standard transformation in all biotic analyses since this is generally acknowledged to allow for balance of rare and common species.

One-way and Two-way nested Analysis of Similarities (ANOSIM), another non-parametric permutation procedure, was applied to the (rank) similarity matrix underlying the MDS ordination to test for statistical significance among sites, canyons and groups. Global R results range between –1 and 1 and are interpreted as R = 1 when within-canyon replicates are more similar than any replicates among canyons, and when R = -1 all replicates among canyons are more similar to each other than to within canyon replicates. However, as with univariate tests, where there are many replicates at each canyon, R may be significantly different from zero and still be inconsequentially small. When comparing differences between pairs of samples, in cases where a maximum number of only 10 permutations is possible (as was the case in some analyses), division of each pair can only have, at best, a statistical significance of 10% (or higher if fewer permutations are possible). Comparison of the R values for each pair, rather than the statistical significance level is, therefore, more valuable for analysis in such cases (Clarke and Warwick 1994).

Univariate testing for statistical significance between canyons receiving high or low visits was also made with the STATISTICA software package, using a three-factor Nested Repeated Measures ANOVA design (Factor 1, 'visits'; Factor 2, 'canyon nested in visits'; Factor 3, 'time of sampling'), following the model shown in Table 12.7, p.405, in Underwood (1997), and using the approach taken by Keough and Quinn (1998). The ANOVA analysis was based on the triplicate samples of the faunal data set for the three sampling sessions (spring, summer and autumn) in 2000/2001: all samples within the same canyoning 'season'. Cochran's test for homogeneity of variances was performed before analysis and, if significant, data were transformed. The Repeated Measures Analysis separated the among canyon variance from the within canyon variance. 'Visits' was tested as a main effect against the 'among canyon' term (between subjects in Repeated Measures terminology). Tests of 'time of sampling' and 'visits' x 'time of sampling' were tested against the 'visits' x 'canyon (within canyons)' term, which must be assumed to be an estimate of the residual error. The residual error could not be independently estimated because Repeated Measures sampling was used (Underwood 1997).

In *F*- test analysis against the 'within-canyons' term, the Repeated Measures ANOVA relies on further assumptions about the variance and covariance of the within canyons (or within subjects) term. These are the assumptions of circularity and compound symmetry (Scheiner and Gurevitch 2001). Violation of the assumptions of circularity and compound symmetry can lead to inflated *F*-values for the within-canyons test. Adjusted *F*-values are sometimes used to guard against increased risk of Type I error for within subject *F*-tests. These adjusted *F*-values were not available in STATISTICA. An alternative analysis is a separate analysis for each 'time of sampling', with Bonferroni correction applied. Repeated Measures was retained because the test of main interest, 'visits', was not affected by inflated *F*-values.

Testing for statistical significance between high and low visit sections within two canyons (Du Faur and Rocky Creek) was also made using the same analytical approach as detailed above but a three-factor Factorial Repeated Measures ANOVA design (Factor 1, 'visits'; Factor 2, 'canyon'; Factor 3, 'time of sampling') was employed. 'Visits' and 'canyon' were considered to be orthogonal to each other.

Water quality in the canyons was also assessed to investigate potential association with different levels of human visitation. A weighted version of the SIGNAL biotic index (Stream Invertebrate Grade Number Average level) described by Chessman (1995) was employed (SIGNAL95-W). This index was calculated by multiplying the square root of the abundance of each family present in the full faunal data set by its SIGNAL grade, summing the products, and dividing by the sum of the square root abundance data. The resulting SIGNAL95-W index ranges from 0 = lowest quality to 10 = highest quality. A Repeated Measures Three-factor Nested ANOVA (Factor 1, 'visits'; Factor 2, 'canyon nested in visits'; Factor 3, 'time of sampling'), using data pooled by canyon and canyons as replicates, was performed using the STATISTICA software package, to test for statistical significance among high and low visit canyon groups for the three times of sampling, spring, summer and autumn of 2000/2001. Testing for statistical significance was also made within high and low visit sections of Rocky and Du Faur canyons.

Spate events were estimated, based upon the senior author's extensive field notes taken during the study, aborted sampling due to high water, and comparison with rainfall data. Based on this information, canyons were spate-disturbed twice in October 1999, but not in October 2000. Spates occurred at least once during the December 2000/January 2001 sampling period and again at the end of January 2001. The canyons were also disturbed in March in both years: at least once in March 2000 and severely in March 2001. Disturbance was more frequent and severe in Year 2 than Year 1. Overall, sites would have been generally subjected to the same meteorological conditions because of their close proximity (< 20 km). For this reason spate effects were assumed to be equivalent in the eight canyons.

Results

Macroinvertebrate assemblages

In total, 5508 animals were collected, spanning 12 orders, 38 families and 78 RTUs (Table 2). Most families (84.2%) were represented by <2 RTUs, and 47.4% by one. Family richness was concentrated in few orders: 84.2% were represented by 50% of orders: Trichoptera (10), Diptera (6), Coleoptera (5), Ephemeroptera (4), Plecoptera (4) and Odonata (3). The other six orders were represented by a single family. Ephemeroptera was the most abundant order (42.3% of all animals) compared with Diptera (18.6%) and Coleoptera (13.0%), while the Leptophlebiidae (Ephemeroptera) was the dominant family (35.4% of all animals), compared to the Chironomidae (Diptera) 14.1%, Elmidae (Coleoptera) 7.3%, Coloburiscidae (Ephemeroptera) 6.3% and Aeshnidae (Odonata) 5.1%. None of the other 33 families contributed more than 4% towards total abundance. Although not quantified, a wide range of body sizes in the leptophlebiids, representing early to late instars, was evident in every sampling period.

Influence of visitation level on macroinvertebrates within canyons

Analysis of macroinvertebrates showed that there was no significant difference in assemblage composition between canyons with high and low visitation in any sampling period (Table 3). In contrast, a significant difference was observed among individual canyons, and there were low to moderately significant Global R values (range 0.35 – 0.68, significant at <0.3%; Table 3). These differences were not associated with level of visitation, and were not consistent among sampling events. Substantial variability in macroinvertebrate assemblage composition within individual canyons was also evident. These patterns were not consistent among seasons or with visitation level.

A comparison between high and low visitation within the same canyons (i.e., sections of Du Faur Creek and Rocky Creek) also showed that there was no consistent pattern of macroinvertebrate distribution. On occasions there were differences within and between the two canyons but not on other occasions (Table 4). For example, although having the highest overall visitation of any canyon surveyed, and greatest within-canyon difference between high and low visit sections (0 vs 99 mean weekly visits, Hardiman 2003), overall Rocky Creek had similar macroinvertebrate assemblages (i.e., low, non-significant pairwise R values in three of the four sampling periods; 0 - 0.19; > 20% significance; Table 4). In contrast, Du Faur Creek, which had a substantially lower within canyon difference between high and low visitation (0 vs 28 visits), the reverse was true (0.41 - 0.89; 10% significance; Table 4).

There was no significant difference within season (spring 2000, summer 2000/2001, autumn 2001) when invertebrate assemblages were compared among canyons. This showed that visitation did not have a statistically significant impact due to high and low traffic: between canyon types or high and low visit to sections of Rocky and Du Faur creeks. The null hypothesis that there is no impact on macroinvertebrate assemblage composition between canyons receiving high and low visit levels was, therefore, supported.

Influence of water quality on distribution of macroinvertebrsates within canyons

Water quality was 'high' (SIGNAL 95 range = 6.15 - 7.71) across all canyons in all seasons sampled. Human recreation was not shown to have a measurable significant impact on water quality at current recreation levels.

Table 2. Total number and percentage in each category (order, family and Recognisable Taxonomic Unit) of macroinvertebrates collected from 30 sites in 8 canyons in the Blue Mountains between 7 October 1999 and 26 March 2001 (n = 5503 individuals).

Taxon	Number	%	Taxon	Number	%	Taxon	Number	%
Order			**Family (continued)**			**RTUs (continued)**		
Ephemeroptera	2329	42.3	Gordiidae	4	0.1	Niscigastridae 1	8	0.1
Diptera	1026	18.6	Odontoceridae	4	0.1	Polycentropidae 1	7	0.1
Coleoptera	718	13.0	Dytiscidae	3	0.1	Calocidae/Helicophidae 2	6	0.1
Plecoptera	538	9.8	Leptoceridae	3	0.1	Tipulidae 4	6	0.1
Trichoptera	424	7.7	Philorheithridae	2	0.0	Aeshnidae 2	5	0.1
Odonata	308	5.6	Empididae	1	0.0	Athericidae 1	5	0.1
Neuroptera	133	2.4	Limnocharidae	2	0.0	Ceratopogonidae 4	5	0.1
Decapoda	18	0.3	Pyralidae	1	0.0	Grinidae 1	5	0.1
Megaloptera	8	0.1				Leptophlebiidae 3	5	0.1
Gerdioidea	4	0.1	**RTUs**			Tipulidae 6	5	0.1
Acarina	1	0.0	Leptophlebiidae 1	1931	35.1	Conoesucidae 1	4	0.1
Lepidoptera	1	0.0	Chironomidae 3	520	9.4	Gordiidae 1	4	0.1
			Elmidae 1	392	7.1	Odontoceridae 1	4	0.1
Family			Coloburiscidae 1	274	5.0	Ceratopogonidae 1	3	0.1
Leptophlebiidae	1949	35.4	Aeshnidae 1	274	5.0	Chironomidae 9	3	0.1
Chironomidae	778	14.1	Eustheniidae 1	219	4.0	Gripopterygidae 4	3	0.1
Elmidae	402	7.3	Hydropsyclidae 1	171	3.1	Baetidae 2	2	0.0
Coloburiscidae	346	6.3	Psephenidae 1	166	3.0	Chironomidae 6	2	0.0
Aeshnidae	279	5.1	Scirtidae 1	140	2.5	Dytiscidae 1	2	0.0
Eustheniidae	219	4.0	Hydrobiosidae 1	132	2.4	Gripopterygidae 5	2	0.0
Hydropsychidae	172	3.1	Neurorithidae 1	131	2.4	Leptoceridae 1	2	0.0
Psephenidae	166	3.0	Chironomidae 4	125	2.3	Neurorthidae	2	0.0
Gripopterygidae	144	2.6	Austroperlidae 1	121	2.2	Polycentropidae 2	2	0.0
Scirtidae	140	2.5	Chironomidae 1	96	1.7	Chironomidae 7	1	0.0
Hydrobiosidae	133	2.4	Gripopterygidae 3	93	1.7	Chironomidae 8	1	0.0
Neurorthidae	133	2.4	Simuliidae 1	87	1.6	Dytiscidae 2	1	0.0
Tipulidae	129	2.3	Tipulidae 1	71	1.3	Ecnomidae 2	1	0.0
Austroperlidae	121	2.2	Notonemouridae 1	54	1.0	Elimidae 3	1	0.0
Simuliidae	88	1.6	Tipulidae 2	45	0.8	Empididae 1	1	0.0
Notonemouridae	54	1.0	Calocidae/Helicophidae 1	36	0.7	Gyrinidae 2	1	0.0
Calocidae/Helicophidae	42	0.8	Gripopterygidae 1	29	0.5	Gyrinidae 3	1	0.0
Philopotamidae	29	0.5	Philopotamidae 1	29	0.5	Hydrobiosidae 2	1	0.0
Baetidae	26	0.5	Ecnomidae 1	25	0.5	Hydropsychidae 2	1	0.0
Ecnomidae	26	0.5	Baetidae 1	24	0.4	Leptoceridae 2	1	0.0
Ceratopogonidae	25	0.5	Chironomidae 2	22	0.4	Leptophlebiidae 4	1	0.0
Parastacidae	18	0.3	Parastacidae 1	18	0.3	Leptophlebiidae 7	1	0.0
Corduliidae	16	0.3	Ceratopogonidae 2	17	0.3	Leptophlebiidae 8	1	0.0
Amphipterygidae	13	0.2	Corduliidae 1	16	0.3	Limnocharidae 1	1	0.0
Polycentropidae	9	0.2	Amphipterygidae 1	13	0.2	Philorheithridae 1	1	0.0
Corydalidae	8	0.1	Gripopterygidae 2	12	0.2	Philorheithridae 3	1	0.0
Moscogastrodae	8	0.1	Leptophlebiidae 5	10	0.2	Pyralidae 1	1	0.0
Gyrinidae	7	0.1	Elmidae 2	9	0.2	Simuliidae 2	1	0.0
Athericidae	5	0.1	Chironomidae 5	8	0.1	Tipulidae 5	1	0.0
Conoesucidae	4	0.1	Corydalidae 1	8	0.1	Tipulidae 7	1	0.0

The influence of environmental structuring on macroinvertebrate assemblages

Analyses of water quality parameters tested to investigate the possible causes of the observed biotic patterns failed to explain such variability. All streams were within ANZECC (2000) guidelines for upland streams of New South Wales: they were clear, cool and varied between acidic (when influenced by bogs) and alkaline, high in dissolved oxygen and conductivity (Table 5). Comparison of the biological data with physicochemical data failed to reveal any significant correlation.

Discussion

Contrary to the expectations of the land management agency for this wilderness area (see e.g., NPWS 2001), there was no apparent impact on macroinvertebrate assemblages or water quality at current levels of recreational use. Even within Rocky Creek, the canyon with the highest recorded level of visitation, there was no indication that recreation was resulting in a press or pulse disturbance. Such lack of apparent impact was observed even in mid-summer, the time of highest recreational activity (Hardiman 2003), when the aquatic environment is considered to be naturally at its most stressed due to the relatively warm water temperatures, low water level and low flow rates compared to other times of the year (*cf*. Kuss *et al*. 1990). Overall, there was no evidence of pulse disturbance at any one period, ramp disturbance increasing within season, or press disturbance across the seasons sampled.

As previously noted, the canyons' location in a remote wilderness setting provides protection from anthropogenic effects of human activity including logging, which have been shown to be detrimental to macroinvertebrate assemblages in other streams, in Australia and internationally. In the same upland region, Wright and Burgin (2009) found dramatic changes in macroinvertebrate abundance and species richness in communities exposed to organic or heavy metal pollution compared to background

Table 3. Two-way Nested Analysis of Similarities (canyon nested within high/low visit group x 3 replicates/sample) of macroinvertebrate assemblages between eight individual canyons and high/low visit groups. Data square root transformed.

Time of sampling	Test	Global R	Level of significance (%)
Spring 1999	Among canyons	0.35	0.1
	High vs low visit groups	0.20	11.4
Autumn 2000	Among canyons	0.35	0.3
	High vs low visit groups	0.002	48.1
Spring 2000	Among canyons	0.68	0.1
	High vs low visit groups	-0.20	91.4
Summer 2000/2001	Among canyons	0.46	0.1
	High vs low visit groups	-0.13	79.0
Autumn 2001	Among canyons	0.51	0.1
	High vs low visit groups	-0.12	79.5

Table 4. One-way Analysis of Similarities of macroinvertebrate assemblages of high and low visit sections within Rocky Creek and Du Faur Creek. Data square root transformed.

Figures adjoining canyon letter denote mean weekly visits	0 vs 28.0 Du Faur Creek		0 vs 99.4Rocky Creek	
Time of sampling	Pairwise	Significance (%)	Pairwise	Significance (%)
Autumn 2000	0.89	10	0.19	20
Spring 2000	0.78	10	0.07	50
Summer 2000/2001	0.41	10	0.00	50
Autumn 2001	0.15	50	0.52	10

Table 5. Summary of physico-chemical results for eight canyons (four with high and four with low visitor numbers) sampled in the Blue Mountains over five sampling periods between October 1999 and March 2001 (All parameters in all seasons over time where within ANZECC2000 guidelines for upland streams of New South Wales).

Physio-chemical parameter	Mean	Maximum	Minimum
Turbidity (NTUs)	<10	<10	<10
pH	5.3	11.4	3.8
Conductivity (mu S)	33.4	68.5	13.3
Salinity (ppM)	14.5	31.5	5.4
Temperature (0C)	13.9	17.7	9.7
Dissolved Oxygen (pp - Mg/l)	8.1	11.2	4.9

levels. Natural variability, mediated by local climatic conditions, rather than human activity, is assumed to be the most likely explanation for the observed variation in macroinvertebrate assemblages among the individual canyons of our study.

The steep walled and narrowness of these slot valleys results in high variability in the amount of sunlight that reaches the canyon floor, together with varying underlying rock types, confined 'choke points' of log jams and siltation, canyons typically vary considerably along their length, and this may result in patchy, complex systems. Frequent, unpredictable spates occur as a result of storms in the catchment, and the associated flash flooding may cause extensive scouring of the streambed and re-structuring of canyon topography at both the micro- and meso-habitat scale. Such disturbances, exacerbated by narrow, sheer rock walls and a limited (or absent) hyporheic zone offer, at best, few refugia to the aquatic fauna which are undoubtedly dislodged over considerable areas. Support for this suggestion is provided by Bond and Downes (2000). They studied the less dynamic system of a lowland river in south-east Australia and observed that hydropsychid (caddis fly) density was higher on large/medium sized stones than on small stones. This is consistent with the notion that small stones are likely to be disturbed more often than large stones. They also found, however, that even large rocks, fixed in the streambed, gave no refuge to organisms during large spate discharges. Under these circumstances there is apparently sufficient shear stress to abrade hydropsychids from surfaces, regardless of level of substrata disturbance. In contrast to large lowland rivers, where the onset of spates is relatively more gradual and predictable, or other streams with seasonally predictable spate patterns (Rosser and Pearson 1995; Brewin *et al.* 2000), macroinvertebrate densities in streams subject to spates that are sudden and unpredictable have been shown to decline dramatically following spate disturbance in tropical (Flecker and Feifarek 1994), temperate (Brooks and Boulton 1991; Robson 1996; Bond and Downes 2000) and arid (Molles 1985) regions. Once disturbed by such unpredictable changes in water level, the time since the last disturbance and time until the next is probably a fundamental factor in the structuring of the native invertebrate assemblages. Greater frequency of such spates possibly leads to adaptation in the aquatic fauna in the form of increased resilience compared to aquatic systems subject to lower disturbance regimes. Although evidence for such adaptation is scarce, some support is provided by the findings of Matthaei *et al.* (1996) who studied a Swiss pre-alpine river and found higher faunal resilience at a site subject to higher disturbance frequency within a background of relatively lower disturbance frequency. An alternative explanation is that only species capable of coping with substantial disturbance live in such habitats.

As a result of exposure to such unpredictable, wide-reaching natural disturbances over geological time, aquatic invertebrates in the Blue Mountains canyons are likely to have evolved mechanisms to become highly resilient to such natural disturbances, probably in the form of high mobility, fast recolonisation traits and/or year round breeding. Support for this hypothesis in the current study comes from the overwhelmingly dominant abundance of the mayfly order Ephemeroptera (42.3% of the animals collected), itself dominated by the family Leptophlebiidae, the most common of Ephemeropteran families in Australia (Williams 1980; Chessman 1986). Mayfly nymphs are highly mobile, substratum generalists (Merritt and Cummins 1996) and have been shown to rapidly colonise (Brooks and Boulton 1991). They may attain pre-disturbance densities within one day post-disturbance (Hardiman and Burgin, 2011). The presence of Leptophelbiidae in a range of instars in all seasons of the study, and the lack of a pronounced pattern of seasonality in this family, or the aquatic fauna as a whole, further support the observation of others (Hart 1985; Death and Winterbourn 1995) of a flexible, opportunistic fauna, adapted to environmental uncertainty.

If streambed invertebrates inhabiting canyons are naturally adapted to frequent, natural physical disturbance, it seems likely that additional anthropogenic physical impact, in the form of trampling by canyoners as they pass through the canyons, would represent negligible additional impact, even if applied over extended periods of time, and in relatively high volume. These observations contrast with human trampling impacts in other ecosystems. For example, Keough and Quinn (1998) found disturbances on a marine foreshore environment caused a series of pulse responses (recovery following disturbance) in two geographically close sites but in a third, more distant site, periodic disturbance over two years resulted in a long-term press response for the next four years. Ross (2006) found trampling effects in a temperate mangrove forest still persisted for at least two years after trampling ceased. In contrast, we did not observe a response to impact (ramp, pulse or press) due to canyoner trampling, even within canyons with high visitation, despite each having been subject to relatively high levels of human trampling for at least the previous 10 years.

Macroinvertebrate assemblage recovery may also be influenced by the time of year at which the human impact occurs. In a study in the Acheron River in temperate southeastern Australia, Doeg *et al.* (1989) observed that species density and abundance took substantially longer to recover in winter than in summer. Human traffic through canyons is inconsequentially low in winter due to the low air and water temperature (Hardiman 2003). However, there were also no consistent differences in invertebrate assemblage composition over seasons sampled between areas with high and low visitation, within or between canyons. This suggested that there were no substantial differences in faunal recovery rates between seasons. These conclusions are supported by the study of Wolfenden (2005) in another highly visited canyon in the same area (Claustal Canyon). He observed that within one month of an extreme spate, the composition of the invertebrate assemblage was equivalent to that observed before the spate.

The data reported here are the first comparison of different trampling frequencies and intensities between seasons and years undertaken in one of the most unpredictable of Australian lotic environments. We suggest that, at current levels of visitor trampling, the impact on canyon streambed invertebrate assemblages is inconsequential compared to natural perturbations in these slot valleys. This contrasts with the research on other forms of aquatic human recreation such as canoeing, which has been shown to disrupt or cause local extinction of streambed invertebrates in shallow zones of rivers (Liddle and Scorgie 1980), or walking in a range of ecosystems including marine foreshores and mangrove forests, which have been demonstrated to produce sustained impacts (Keough and Quinn 1998; Ross 2006).

Current levels of canyoning visits also have had no discernible impact on water quality (*cf*. ANZECC 2000), which remained at conditions that would be considered pristine throughout all canyons, at all sampling visits, and was equivalent between high and low visit canyons.

Increase in siltation in canyon streams from canyoners stirring up substrata and/or bank erosion also had no measurable effect on the macroinvertebrate assemblages. Bond and Downes (2003) studied benthic macroinvertebrate fauna in eight artificial streams located in upland southeast Australia. They introduced fine sediment, alone and in combination with simulated small spates, and observed that there was minimal impact on the fauna. These data are unique in paralleling our findings.

The conclusion drawn from our data is that, based upon arguably the most widely recognised bioindicators of aquatic health (i.e., benthic macroinvertebrate assemblage composition; water quality), current levels of canyon recreation have not had a measurable impact on canyon ecosystems that have been traditionally viewed as fragile. We conclude that at the current level of recreational use, even in the most popular canyons, the sport is sustainable at present levels of canyoning, at least in terms of water quality and macroinvertebrates communities.

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